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16. ABSTRACT
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EFFECTS OF ACID PRECIPITATION IN NORTH AMERICA

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Recent evidence indicates that acid rain is a growing environmental phenomenon of potentially far reaching consequences and increasing geographical extent in North America. Acid rain is but one aspect of the broader problem of atmospheric deposition which includes snow, fog, and dry deposition of material. First noticed and studied in the Scandinavian countries, acid precipitation has now been well documented in the United States, first in the northeast and more recently throughout much of the United States east of the Mississippi River. Numerous streams and lakes in regions with poorly buffered soils have become devoid of fish, have an impoverished aquatic flora and fauna, and are changing toward conditions of low aquatic productivity. Evidence also indicates that acid precipitation may cause damage to forest growth, crop production, wildlife, and man-made materials such as buildings, metals, paints, and statuary.

Introduction

Recent reviews of available data (Cogbill and Likens, 1974; Nisbet, 1975; Likens and Bormann, 1974) indicate that precipitation in a large region of North America is acidic when compared with the expected *p*H value of 5.6 for pure rain water in equilibrium with CO_2 (Newman, 1975; Galloway *et al.*, 1976). It has also been shown that the time course of change in *p*H of precipitation from the mid-1950s to the mid-1970s in the northeastern United States and Canada has been dramatic (Likens, 1976). Moreover, acid precipitation has spread southward and westward in the United States (Cogbill and Likens, 1974; Likens, 1976). More recent information indicates that in the southern and western portions of the United States *p*H values between 3.0 and 4.0 are observed during individual storms (Likens, 1976).

Although prior to 1955 the historical record on changes in acidity of precipitation is sparse, there are data which indicate that by the mid-1950s precipitation in the eastern United States was already acidic, and that the acidity of rain and snow in that region increased significantly sometime between 1930 and 1950. Numerous lakes which were populated with fish prior to the 1950s are now devoid of fish or have fish stocks which have older age distributions or missing year classes in young age groups.

A growing body of evidence suggests that acid rain is responsible for adverse effects on the public welfare. Such effects include the acidification of lakes and rivers with resultant damage to fish and other components of aquatic ecosystems, acidification and demineralization of soils, possible reductions in crop and forest productivity, and deterioration of man-made materials (Preston

and Sanyal, 1956; Glass, 1978; Glass *et al.*, 1979; Dochinger and Seliga, 1976). These effects can be cumulative or may result from peak acidity episodes (Berry and Bachman, 1977).

A drop in the *p*H of precipitation has been observed for many years in Scandinavia (Barrett and Brodin, 1955). A monitoring network there has shown that since the mid-1950s precipitation in northwestern Europe has increased in acidity and that this acidity is currently widespread geographically. The hydrogen ion concentration of precipitation in some parts of Scandinavia has increased more than 200-fold during the past two decades (Almer *et al.*, 1974).

Data from New York State and other parts of New England indicate that approximately 60% to 70% of the acidity is due to sulfuric acid and 30% to 40% of the acidity is due to nitric acid. These strong acids are thought to stem largely from gaseous man-made pollutants such as sulfur oxides and nitrogen oxides produced primarily, although not exclusively, from the combustion of fossil fuels. The relative proportion of nitric acid derivatives and sulfuric acid derivatives may be an adequate indication of the nature of the source from which the acid rain was derived—a high proportion of oxides of nitrogen or of nitric acid derivatives would suggest automobile or mobile sources whereas a high proportion of sulfuric acid derivatives would indicate stationary sources such as power plants, smelters, and heavy industry. It is interesting to note that in the early part of this century precipitation acidity in the vicinity of Leeds, England (a heavy coal use region), was approximately 75% attributable to sulfur compounds (Crowther and Ruston, 1911) and *p*H in rain and fog appears to have dropped below 3.0 on occasion.

Emission sources of sulfur oxides and nitrogen oxides are widely distributed within and outside urban centers. Contributions can come from both lower and higher height stacks and from near ground level sources. Sulfates are present in the stack gases associated with coal-fired and oil-fired sources. The amounts of sulfuric acid and other sulfates found in plumes can be sufficient to affect plume opacity and contribute to the fallout of acid particles near the source. In plumes from elevated sources, lack of contact with the ground tends to preserve precursors for some distance downwind. Especially at night and in the early morning hours ground-based inversions can isolate the plume aloft so that near source deposition is minimized (Smith *et al.*, 1978).

The urban plume already contains organics, sulfur oxides and nitrogen oxide precursors to sulfates and nitrates (Fennelly, 1976). Photochemical atmospheric reactions can form oxidants, sulfates and nitrates as the urban plume progresses downwind. During periods of effective photochemical activity, however, urban plumes will tend to be well mixed all the way to the ground. Therefore, dry deposition processes are competing with atmospheric reactions as sinks for sulfur oxides and nitrogen oxides (Smith *et al.*, 1978).

In North America, acid precipitation is considered to be most severe in the northeast. However, recent data show the geographic extent of the problem to be increasing in the southeast and midwest with all states east of the Mississippi affected to some degree. Furthermore, there is recent evidence of acid rain in Alberta and in the western United States at least in major urban centers such as the Los Angeles area, San Francisco and Seattle (Liljestrand and Morgan, 1978; McColl and Bush, 1978). The ratio of sulfur derivatives to nitrogen derivatives (approx. 1/2) indicates that acidity in and near urban areas of the west is probably due to automobiles rather than to stationary sources. Unlike more conventional atmospheric pollution, that giving rise to acid rain may not exceed air quality standards nor cause immediately obvious damage to receptor organisms and materials.

Canadian Emissions

Emissions of SO_2 in Canada vary somewhat according to smelter activity, but in 1978 totalled 5.0×10^9 kg. These have given cause for concern in four main areas and several smaller ones. Major areas are the Sudbury region of Ontario (14×10^3 km^2 with an emission of 1.4×10^9 kg), the Windsor–Sudbury–Quebec City triangle in southern Ontario and Quebec (200×10^3 km^2), southwestern Alberta (80×10^3 km^2 with an emission of 0.6×10^9 kg), and an area common to northern Alberta and Saskatchewan downwind from the tar sands (100×10^3 km^2 with an emission of 0.1×10^9 kg). The isolated centers include Rouyn-Noranda and Murdochville, Quebec (0.6×10^9 kg), and Flin Flon and Thompson, Manitoba (0.5×10^9 kg). In one of the major areas—Sudbury—high ambient concentrations of SO_2, acting with acid precipitation and particulate matter, have for many years been a major threat to vegetation, soils, and other environmental values. In southern Ontario and Quebec local emissions, combined with SO_2 and NO_x emissions in surrounding areas, have exceeded the normal resilience of water bodies and some agricultural crops. Emissions from sour gas processing and tar sands extraction in Alberta have not so far posed serious threats but the sensitivity of certain of the agricultural and forest crops of the two areas and the ease with which sulfur–selenium imbalances in animal and wildlife nutrition could be brought about, do give rise to concern (Statistics Canada, 1979; Massey *et al.*, 1977; Rennie and Halstead, 1977).

U.S. Emissions

The comparable SO_2 load for the United States is approximately 38×10^9 kg/yr. However, emission rates from U.S. versus Canadian sources do not necessarily

indicate net transboundary flux or deposition rates. Major source regions of the United States include the Ohio and Tennessee Valleys, the north-central and northeast industrial areas, and the Washington, DC, to Boston urbanized corridor.

Data from the Canadian precipitation sampling and analysis network (CANSAP) show that large parts of eastern Canada, stretching from the Manitoba–Ontario border to Newfoundland, are receiving substantial amounts of sulfate in the form of wet deposition (Whelpdale and Galloway, 1979). Moreover, in the more markedly affected parts of south-central Ontario and Quebec, the amounts being deposited range from 18.0 to 78.0 kg of $SO_4^=$ per hectare per year. Such deposition rates are somewhat less than those being received in the more severely affected parts of the northeastern United States, but they are of the same general order as those recorded in Scandinavia where serious damage to some environmental components has occurred (Braekke, 1976).

Soils and Geology

Serious though these emission and deposition data appear, it must be recognized that they can only begin to take on a degree of significance when the nature and properties of the impacted materials are also taken into account. For this reason, precipitation would have to become very acid indeed in the Canadian prairies to generate environmental concern for calcareous or sulfur deficient soils (Halstead and Rennie, 1977). The same holds for the calcareous and neutral soils of southern Ontario, in spite of appreciable sulfate loadings. However, for most of eastern Canada, soils are podzolic, not well endowed with nutrient elements and display natural acidities ranging from about *p*H 4 in the surface horizons to around 5.5 in the lower parts of the profile (Rennie, 1978a).

In general, then, the picture for much of eastern Canada is of a bedrock geology with surficial deposits giving rise to acidic soils and water bodies that are naturally low in alkalinity and calcium reserves. Their neutralizing ability is limited and their resistance to further acidification is tested when the strongly dissociated sulfuric and nitric acids brought in by acid rain supplant a system normally stabilized by the less strongly dissociated mixture of carbonic and other organic acids.

However, such terrestrial and aquatic systems must not be thought of as being unproductive and of little ecological or economic significance. For the eastern Canada forest resource alone, for example, the direct value (after processing) is on the order of $4 billion per year. This forest resource also has indirect and intangible values in providing recreation, habitat for wildlife, stabilizing river flow, preventing erosion and the siltation of water bodies, and in furnishing aesthetic appeal, which are all inestimable. Forest growth rates are not as high as in other more favored soil and climatic zones of North America, but this merely means that larger areas have to be more soundly managed than elsewhere.

Increased Coal Use

The picture for much of the eastern U.S. is one of a patchy mosaic of sensitive and nonsensitive soils and bedrock geology (McFee, 1980; Hendrey *et al.*, 1980; Glass *et al.*, 1980). Concentration of acid sensitive, low buffering capacity bedrock and soils is seen in New England and the Appalachian plateau areas (Galloway and Cowling, 1978; Glass *et al.*, 1980), and in the upper midwest.

As the United States and Canada place increasing reliance on coal as an energy source due to crude oil prices, foreign crude oil supply problems, and public questions about the safety of nuclear power, air pollution emissions from energy producers will increase (Rennie, 1980a).

Switching fuel from natural gas or oil to coal will make the task of reducing sulfur emissions from new and existing power plants difficult. Mobile and stationary sources of NO_x will also continue to contribute to the overall loading of acidity to the aquatic and terrestrial environment. In brief, it seems probable that acid deposition to the environment will at least remain at present levels and might be expected to increase until the end of this century.

Watershed Sensitivity—Lakes and Streams

As more coal-fired electric generating plants are built and become operational, especially in areas that are "upwind" in the airshed of the west-central United States, the Midwest, and the Southeast, the quantities of acid forming substances will exceed the acid-neutralizing assimilative capacities of watersheds and the processes of lake and stream acidification in many pristine areas of North America will be exceeded. The beginning point or "threshold" value for this process for Swedish lakes has been determined to be where the quantities of acids in the atmosphere cause the annual average *p*H of precipitation to be depressed to 4.6, a value which corresponds to a hydrogen ion concentration ten times that of natural rain (Wright, 1976). Increased acid doses beyond the "threshold" value have resulted in the acidification of more than 15,000 lakes in Sweden and Norway, most to extreme acid levels where all fish have become extinct.

The transferability of data from Scandinavian observations to North American watershed's responses to acidified precipitation that interact with acids and related components will depend on the similarity of the environmental factors of the watersheds, and primarily on the *lack* of acid neutralizing (or alkaline) soils. This

predominant factor places limits on the geographic regions where watersheds are susceptible to damage from acid precipitation. Lakes in the Adirondack State Park of New York, the Boundary Waters Canoe Area–Voyageurs National Park (BWCA–VNP) of Minnesota, and Nicolet National Forest area of northern Wisconsin are examples of such susceptible environments in the U.S. In the Adirondack area of New York State the "threshold" acid precipitation value has been exceeded for several years and as a result, lake *p*H values are depressed to the point where fish have stopped reproducing and some 100 lakes are now fishless (Schofield, 1976). Portions of eastern Canada have similarly been receiving acid deposition which exceeds the *p*H 4.6 threshold which results in fishless lakes (Beamish, 1976).

In northern Minnesota and Wisconsin, the process of lake acidification is just beginning and only the most susceptible lakes are being affected. This area is one that exhibits the properties of maximum sensitivity to acidification due to extremely soft water, thin and sandy soils, and very sensitive terrestrial and aquatic plants and animals. In addition, the BWCA–VNP is a wilderness area where the U.S. Congress has taken extreme care to preserve the environment against the impacts of pollution and *no* man-induced changes are desired.

The northern Minnesota and Wisconsin region, like the Adirondack Park area, is found on acidic bedrock which is the main criterion for an acid-sensitive classifi-

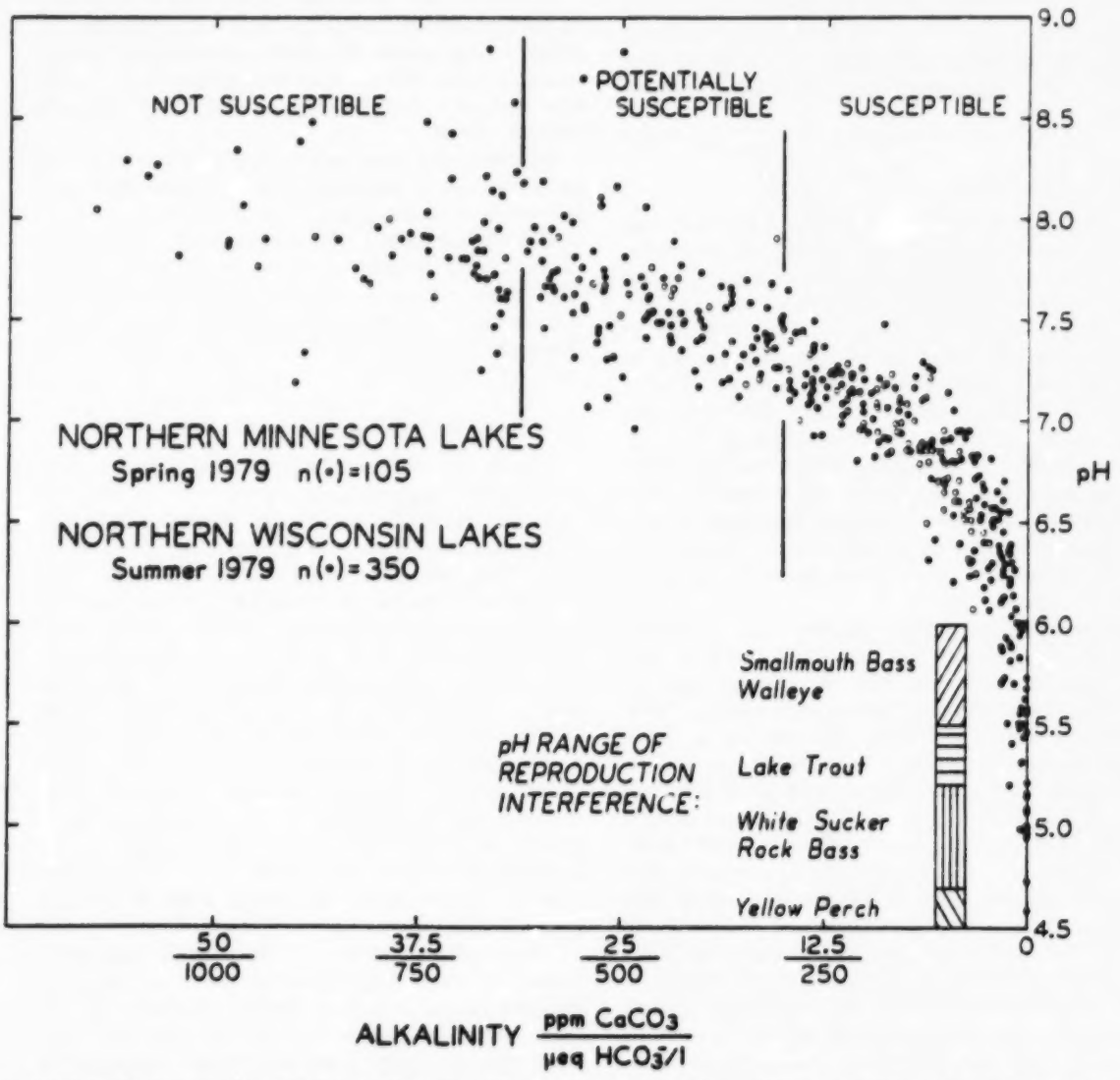

Fig. 1. Relationship between *p*H and alkalinity in 105 northern Minnesota (BWCA-VNP) lakes and 350 northern Wisconsin lakes. The criteria for susceptibility are from Kramer (1976) and *p*H range of reproduction interference to selected fish species are from Beamish (1976).

cation. Of the midwestern states, only upper Michigan, northern Wisconsin, and northern Minnesota have this type of bedrock covering wide areas (Likens *et al.*, 1979). The northern Wisconsin lakes are located in areas of sandy acidic soils and this factor makes them especially acid sensitive (Fig. 1). The lakes in northern Minnesota and Wisconsin have been surveyed only recently for possible impacts from acid precipitation (Glass and Loucks, 1980). About two-thirds of the 455 lakes sampled in 1979 are susceptible or potentially susceptible to change from acid precipitation using the criteria suggested by Kramer (1976). If these lakes are representative of about 20,000 lakes in this region, then the potential for severe ecological damage is serious. The annual average *p*H of precipitation in this region is approximately 4.6 and contributions from increasing emissions, including major coal-fired power plants being built or just completed in the area, will certainly increase the concentrations of acids being added to susceptible watersheds and increase the rate of stream and lake acidification. A larger percentage of lakes in northern Wisconsin are on the verge of acidification and may be receiving higher doses than lakes in northern Minnesota due to their proximity to sources. The potential for adverse effect of acid precipitation is higher because the assimilative capacity has been depleted.

Various fish species have dissimilar responses to different levels of lake *p*H (Beamish, 1976). Figure 1 includes a partial listing of fish species that are vulnerable showing this dissimilarity in reproductive interference at selected *p*H ranges. Of the 350 Wisconsin lakes sampled in 1979, 13% were in the *p*H range where other investigators have found interference to fish reproduction.

Acid dose–Lake *p*H Response

Dose–response relationships between atmospheric loadings of acids to watersheds and the resultant lake *p*H responses have been observed in Sweden (Almer *et al.*, 1978) and are shown in Fig. 2. Since a major source of acid in precipitation is related to sulfur emissions, the extent of lake acidification has been measured as a function of annual atmospheric loadings of sulfate for lakes. The shape of these curves is similar to the acid added *p*H curve for a water sample. These curves were developed by looking at many lakes of similar sensitivities with variable loading. The same relationship is expected for a single lake (watershed) with increasing loadings over time. The sharp drop of this *p*H curve for extremely sensitive lakes shows how a slow increase in "added increments" can and will cause a sharp break in the response curve (Almer *et al.*, 1978; Dickson, 1979). This response is characteristic of very soft water lakes where only a small amount of bicarbonate is available to neutralize the incoming acid; see Fig. 2, curve (1). The *p*H remains relatively constant until all of the carbonate is consumed and then the stronger organic and sulfuric acids control the lake *p*H. In other words, the assimilative capacity (or buffer capacity) of the lake has been exhausted as indicated by the break point and downward slope, especially on curve (1). The response of lakes in less sensitive watersheds follows curve (2). Lakes in the most sensitive bedrock areas have a sulfate loading "threshold" value of 9 kg ha yr (Almer *et al.*, 1978), and also have an additional acid loading due to nitrate. Northern Minnesota receives 4–14 kg ha yr of sulfate with an average load of 11 kg ha yr (Glass and Loucks, 1980).

The shape of atmospheric acid loading-*p*H response curves shown in Fig. 2 is expected for an individual lake as the acid loading from atmospheric sources is increased over time. No information is available concerning the converse of this situation. For a given region where the loading is fairly constant, the *p*H response of lakes may be described as a series of response curves, each reflecting individual differences and characteristics of the particular lake watershed. Some of the factors which control the shape and displacement of these loading-*p*H responses are watershed area–lake volume ratio, soil–geology factors, vegetative cover, groundwater input, organic acid input from bogs, and a host of other factors yet to be defined. The size of watershed and stream order have also recently been discussed as factors in *p*H loading (N. M. Johnson, 1979).

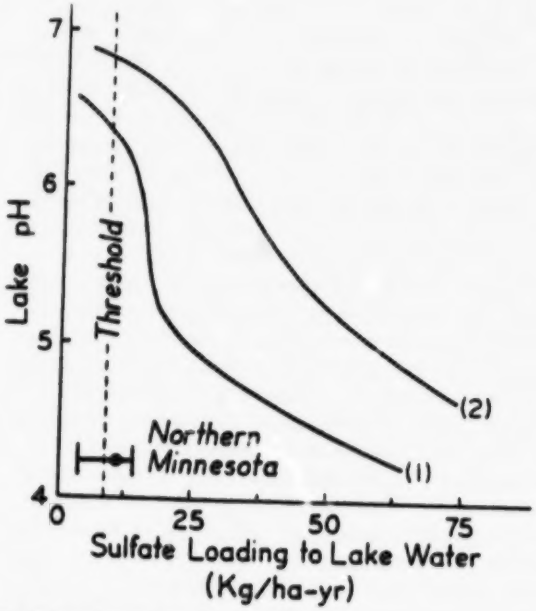

Fig. 2. Relationship of lake *p*H to acid loading, measured as sulfate for Swedish lakes in extremely sensitive surroundings, curve (1) and in somewhat less sensitive surroundings, curve (2) in relation to a threshold of lake acidification (Almer *et al.*, 1978; Dickson, 1979). The sulfate load in northern Minnesota ranges from 4–14 kg/ha/yr with an average of 11 kg/ha/yr. The validity of these response curves as applied to North American watersheds must be verified.

If the acid loading to a particular region is limited to the amounts defined by the upper portions of the response curves for sensitive lake systems, then the magnitude of the damage to the aquatic environment can be minimized. More subtle effects of acid input to lakes have been noticed and additional studies are needed to verify such effects as increased conversion rates of mercury to the more mobile and biologically active organometallic forms (methylmercury) (Dickson, 1979), which lead to greater body burdens in fish.

Acid Precipitation Effects in Unamended Soils and Forests

The federal Canadian Forestry Service has been involved in terrestrial research on air pollution effects for many years beginning in the 1930s, dealing with problems at Trail, British Columbia, and with the separation of pollution effects on forests from those arising from insect and disease attack. Since this time there have been numerous signal contributions to the understanding of pollution problems, usually near strong point emitters (Katz *et al.*, 1939; Linzon, 1971; Sidhu, 1980; Rennie, 1978b; Malhotra and Blauel, 1979; Robitaille, 1979).

Although there are many features common to the problems of acid rain and atmospheric pollutants regardless of how they originate, there are two important differences. First, long-range pollution does not usually exceed conventional air quality standards. Second, it may not cause any spectacular or immediately measurable reduction in tree growth, injury to lesser forest vegetation, or elimination of species (Abrahamsen and Dollard, 1979). A superficial view, therefore, or one not recognizing the injurious effects upon foliage (Jacobson, 1980) that the more extreme acidities of individual rainstorms (Likens and Bormann, 1974) can bring about, would fail to discern much of a problem. However, early research of H. J. Wheeler in Rhode Island and J. A. Voelcker at Woburn, England, may now seem very distant, but their striking demonstration of the soil acidification, calcium removal, and aluminum solubilization effects consequent upon repeated sulfate fertilizer applications should not be forgotten (Russell, 1950).

Figure 3, for example, based on data of Crowther and Basu (1931), shows how repeated sulfate applications made to a slightly acid loamy sand at rates similar to current acid precipitation loadings cause serious losses of exchangeable soil calcium. Significant in these relationships is the greater rate of loss of calcium at the higher calcium levels and the continuation of the accelerated calcium depletion process down to at least soil *p*H 4. The apprehension is that acid precipitation will, in time, bring about analogous effects. Calcium depletion may trigger, in turn, other soil chemical and biological changes and cause a permanent reduction in site quality and tree growth. Although there is appreciable

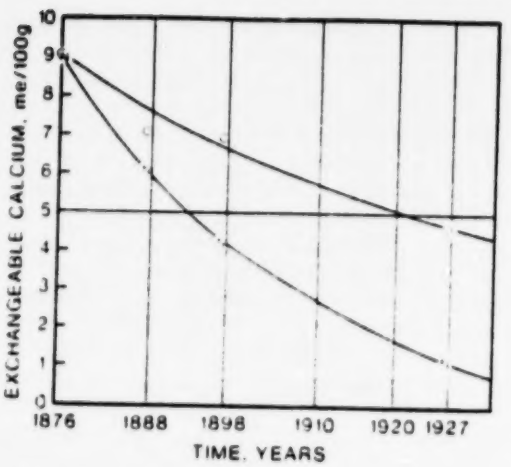

Fig. 3. Depletion of exchangeable calcium from a loamy sand under wheat with (N) and without (C) annual applications of ammonium sulfate at 231 kg/ha/yr (=26 kg/ha/yr of S) (Crowther and Basu, 1931).

literature on the relationships between soil characteristics and *p*H (Alexander, 1980; Rorison, 1980; Schnitzer, 1980) the following specific cases exemplify possible types of chemical and biological changes.

Figures 4 and 5 illustrate the rapid increasing mobilization of the potentially toxic elements, manganese and aluminum, that occurs over the *p*H range of 5.5 to 3.8. The curvilinear relationships plotted from the original data can be expressed by statistically significant equations, while conversions that permit the plotting of either all logarithmic data or all nonlogarithmic concentrations result in statistically significant rectilinear

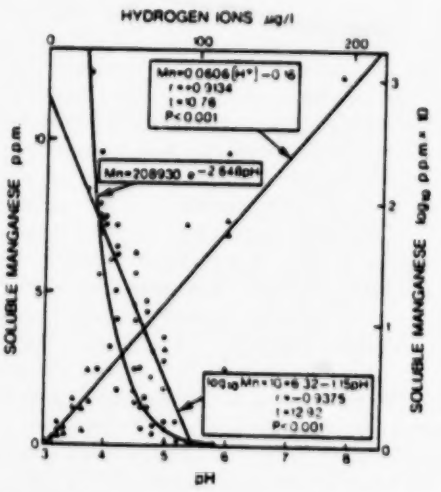

Fig. 4. Solubilization of soil manganese in relation to *p*H in a loamy sand under different fertilizer treatments.

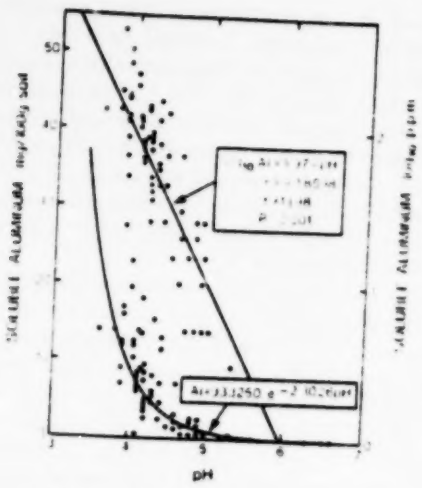

Fig. 5. Solubilization of soil aluminum in relation to *p*H in a Carpathian flish soil (Dobrzanski and Wondrausch, 1953).

relationships. As in the aquatic habitat, so in the soil situation there may be seen a drastic shift in the critically important Ca, Mg:Mn, Al balance.

Similarly, well documented biological cases include reduction in bacterial numbers (Bryant *et al.*, 1979), reduction in earthworm populations (Satchell, 1955), increase in undesirable fungal colonies (Warcup, 1951), reduction in soil arthropods (Strojan, 1978), and a shift in the optimum conditions for mycorrhizal development (Graw, 1979). Moreover, as heavy metal particulates often accompany gaseous pollutants (Phung *et al.*, 1979), the possibilities exist for a variety of noxious elements to be held preferentially on the exchange sites of soil colloids at the expense of the nutritionally useful monovalent and divalent cations (McBride and Blasiak, 1979). These phenomena have already been documented around strong point emitters (Freedman and Hutchinson, 1980).

In spite of these well documented cases it has been said that the more acid forest soils are not sensitive to acid precipitation. In a much published table, Wiklander (1973/74, 1980) claims that acid rain will have only slightly adverse effects upon soils more acid than *p*H 5, and he shows how in the acid rain impingement zone of southwestern Sweden the *p*H gradient of 3.0 to 4.5 down the profile of an iron humus podzol has not changed significantly over the 40 year period 1934–1973.

Although there is clearly much to be researched in the acid rain/acid soil interaction, the point emerging is that this complex area needs to be studied in the context of natural processes and exchange complex reactions (Bache, 1980; Johnson, 1980; Wiklander, 1980). Caution should be exercised when extrapolating from the results of simulation experiments which may or may not approximate natural conditions.

There are several factors of significance to eastern North American forest soils. First, the throughfall precipitation is appreciably enriched in ions by canopy interception and exchange (Cronan and Schofield, 1979). A second factor is a natural biogeochemical nutrient cycle that is influenced by winter freeze-up, a summer soil water deficiency situation of variable intensity, and an early summer period of intense biological activity and soil nutrient assimilation. In this framework the leaching action of a partly neutralized acid solution that may or may not be as acid as the soil horizons it percolates has to be assessed. This is particularly true if the solution is not markedly different in ionic composition from the exchangeable ions held on the soil colloids of already highly unsaturated horizons. It is not surprising, then, for the more acid podzols to be regarded as less prone to degradation (or less sensitive) than the Brunisols. Figure 3 confirms this along with the concept of aluminum and iron buffering in the *p*H 3 to 5 range (Ulrich, 1980).

But what tends to be ignored is the significance of the levels of useful nutrients that could be lost compared with reserves, taking into account the biological system that has to be supported and the cultural practices that are feasible or economic to apply. It is obvious from Fig. 3, for example, that a loss of 0.5 meq/100 g of exchangeable calcium may take longer when a soil has already been degraded to an exchangeable level of 2.0 meq/100 g. Unless the soil has weatherable mineral reserves to replenish this loss the latter may be far more critical for biological productivity than a similar loss in the so-called more sensitive part of the curve around 8.0 meq/100 g.

That highly acid soils can undergo exchange reactions and degradation under the influence of acid precipitation has recently been indicated for the New Jersey Pinelands (A. H. Johnson, 1979). When it is also remembered that forest soils are expected to meet increased nutrient demands of more intensive harvesting and other practices (Voigt, 1979), it is clear that much more precise information is required on the acid precipitation threat.

New approaches to the problem of long-range pollution include "sensitivity mapping." This can be very crude—such as the separation of areas of calcareous and noncalcareous bedrock or the identification of sulfur deficient soils—or it may be large scale and based on a synthesis of soil attributes (McFee, 1980; Glass *et al.*, 1980). Considerable progress has been made in grading the sensitivity of Ontario lakes, using alkalinity as an index, and it is possible that a somewhat similar index based on exchangeable calcium status could be used for soils. For at least the reasons that are explained above, however, such a soil sensitivity index may be no more than a very crude indicator.

Other approaches have been described in detail elsewhere (Rennie, 1980b). Currently in Canada direct effects upon forest and lesser vegetation are being investigated by morphological, physiological, and biochemical means (Malhotra and Sarkar, 1979; Percy and Riding, 1978). Lysimetric studies are in progress, as is work on the relative susceptibility of different forest species to simulated acid rain and pollutant mixtures. In addition, an important interdisciplinary program is bringing together aquatic and terrestrial scientists in three main catchment basins distributed across the primary impingement zone of eastern Canada. It is hoped that by adopting an ecosystem approach, the modelling of the forest basins will reveal qualitative and quantitive information on ionic balances and biological activities as affected by the acid precipitation stress.

Acid Precipitation Effects on Agricultural Systems

There are two basic ways whereby acidic precipitation could impact agricultural crops. First, acidic precipitation can impact directly on the foliar surface of the plant itself causing a direct effect on the leaf or stem of the crop plant. In these cases the most immediate effect would be on crops such as lettuce, spinach, or chard for which the foliage is the valuable portion. Second, acidic precipitation can indirectly affect the crop plant through effects on the soil. For example, changing the *p*H of rainfall which strikes the soil can change the rate at which nutrients are recycled, litter, and other organic matter is broken down through microbial action in the soil, and the rate at which both macronutrients and micronutrients are leached from the soil into surface waters or into groundwaters. Detailed physiological mechanisms whereby these two basic impacts are manifest have been discussed elsewhere (Cowling and Dochinger, 1979).

At *p*H below about 3.0, areas of leaf surfaces become spotted or necrotic. For obvious reasons, vegetables as well as ornamental plants may be unmarketable after exposure to rain or mist at or near *p*H 3. With decreases in functioning leaf area, the photosynthetic process itself must be diminished and the productivity of the plant must necessarily decrease. With changes in aboveground biomass such as would be experienced under low *p*H rain, below-ground biomass would be similarly decreased. It is not yet known what the relationship is between percent leaf area affected (nonfunctional due to necrotic areas) and percent change in growth for plants. However, it is known that, with depressed *p*H, foliar losses of important nutrients such as K, Mg, and Ca are observed (Wood and Bormann, 1975; Fairfax and Lepp, 1975).

Over the long term, nutrient cycling changes in either agricultural land or forestry land can result in lowered fertility due to decreases in essential soil nutrients which are required for normal plant growth.

In three out of five host–parasite systems involving oak trees and kidney beans, alterations occurred in the pathogenicity of the parasite under conditions of artificial sulfuric acid precipitation at *p*H 3.2. An additional observation was that root nodule number and nitrogen fixation of kidney bean were diminished by sulfuric acid rain at *p*H 3.2, and that bean yield was decreased in low cation exchange capacity soils but not decreased in soils of higher (greater than 3 meq 100 g soil) cation exchange capacity (Shriner, 1976).

Rain water at approximately *p*H 4.0 due to volcanic activity in the Kona district of Hawaii has been shown to adversely affect tomatoes. While 5 kg per plant of salable fruit was obtained from tomato plants grown under a plastic rain shelter, no marketable fruit was produced on plants growing immediately outside the rain shelter (Kratky *et al.*, 1974). While it is possible that gaseous pollutants such as SO_2 may also be present near fumaroles or volcanic activity, the fact that the plastic rain shelter had no sides and provided enough protection to the plants to permit fruiting indicates that gases alone are probably not the cause. It is possible that the causative agent was not rainfall, but dry fallout. This possibility should be investigated experimentally. However, it has been shown that cations, plant growth regulating substances, and other materials are leached from growing plants by rainfall (Tukey, 1970) and that leaching rates increase for many materials as *p*H decreases (Hindawi *et al.*, 1980).

While the data relating the effects of acid precipitation to crop yield and production are somewhat sparse, there is considerable indication that acid rainfall is deleterious. In order to pursue this hypothesis somewhat further, an extensive screening program to look at the effect of sulfuric acid rain on virtually every major field crop of the United States was initiated by the U.S. Environmental Protection Agency (Glass *et al.*, 1980; Lee *et al.*, 1980). This study was undertaken in order to determine the sensitivity of crops to simulated acid precipitation in an experimental farm facility because of the potential for widespread economic damage to a number of field crops. Artificial rain at *p*H 3.0, 3.5, 4.0, and 5.5 was applied three times per week during the growing season. Tentative results from this study indicate

- Acid rain damage to foliage is not necessarily reflected in yield (grams dry weight) reductions for the crop;
- Acid rain damage to crops where foliage is the marketable portion such as spinach, chard, or lettuce can diminish crop value;
- Acid rain caused a significant decrease in the yield of five crops, stimulated the yield of six crops and caused no statistically significant yield loss or increase in seventeen crops.

During the 1980 growing season, 25 crops will be screened by exposing them to an acid rain mixture of sulfuric and nitric acids. Yield will be the measured variable of chief concern. Results are expected by the end of 1980.

Conclusion

In conclusion, there is substantial reason to suspect that the deposition of acidic precipitation throughout wide geographic areas of the eastern United States and Canada will have adverse effects on aquatic systems, forests, and agricultural systems. Evidence presented in this paper suggests several avenues of research which should be pursued to further define the magnitude and extent of the effects of acid precipitation on resources. The cumulative threat of acid precipitation is recognized and a concerted attack is being spearheaded by Environment Canada and the U.S. Environmental Protection Agency that brings together different disciplines and jurisdictions. Ongoing pollution studies based on strong point emitters and special new investigations are being applied in a number of promising approaches. These include the use of sensitive lichens as indicator species, previsual biochemical tests on tree and other plant tissues, differential depositional patterns of pollutants on soils, delineation of "sensitive" soils, forest, and water bodies, and comprehensive catchment basin studies. While direct effects on terrestrial and aquatic ecosystems should be investigated intensively, effects on the abiotic components of such systems should also be studied. For instance, a number of processes involving the impact of acidic precipitation on soil systems, both amended agricultural soils and natural forest soil systems, need to be investigated. Processes such as soil litter decomposition, nutrient cycling, leaching of nutrients, toxicants, and other cations and anions from natural and managed systems should be investigated. Further definition of sensitive areas of North America should be accomplished so that field research programs can be focused geographically in those areas where the impact is suspected of being greatest.

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